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Net anthropogenic phosphorus inputs: spatial and temporal variability in the Chesapeake Bay region

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Abstract We estimated net anthropogenic phosphorus inputs (NAPI) in the Chesapeake Bay region. NAPI is an index of phosphorus pollution potential. NAPI was estimated by quantifying all phosphorus inputs and outputs for each county. Inputs include fertilizer applications and non-food phosphorus uses, while trade of food and feed can be an input or an output. The average of 1987, 1992, 1997, and 2002 NAPI for individual counties ranged from 0.02 to 78.46 kg P ha^{-1} year^{-1}. The overall area-weighted average NAPI for 266 counties in the region was 4.52 kg P ha^{-1} year^{-1}, indicating a positive net phosphorus input that can accumulate in the landscape or can pollute the water. Large positive NAPI values were associated with agricultural and developed land cover. County area-weighted NAPI increased from 4.43 to 4.94 kg P ha^{-1} year^{-1} between 1987 and 1997 but decreased slightly to 4.86 kg P ha^{-1} year^{-1} by 2002. Human population density, livestock unit density, and percent row crop land combined to explain 83% of the variability in NAPI among counties. Around 10% of total NAPI entering the Chesapeake Bay watershed is discharged into Chesapeake Bay. The developed land component of NAPI had a strong direct correlation with measured phosphorus discharges from major rivers draining to the Bay ($R^2 = 0.81$), however, the correlation with the simple percentage of developed land was equally strong. Our results help identify the sources of P in the landscape and evaluate the utility of NAPI as a predictor of water quality.

Keywords Anthropogenic · Budgets · Nutrients · Phosphorus · Watershed

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Introduction

Eutrophication has long been a major focus of limnology and marine research (Vollenweider 1968; Hutchinson 1969; Likens 1972), but has gained in prominence in recent decades as anthropogenic modification of the landscape continues (Smetacek et al. 1991; Nixon 1995; Jordan et al. 1997; Howarth et al. 2000; Rabalais et al. 2001; Boyer et al. 2002; Kemp et al. 2005; Alexander and Smith 2006). Human activities, such as agriculture and urban development, have been linked to increases in nutrient and sediment loading (Correll 1987; Turner and Rabalais 1991; Carpenter et al. 1998) and can adversely affect water quality, productivity, and trophic structure (Ryding and Rast 1989; Rekolainen et al. 1995; Rast and

Thornton 1996) in receiving lakes, wetlands, estuaries, and coastal waters (Orth and Moore 1983; Officer et al. 1984; Jordan et al. 1991a, b; Frink 1991; Golterman 1995; Howarth et al. 1996; Vukadin et al. 1996; Carpenter et al. 1998; Boesch et al. 2001). Coastal watershed eutrophication has become a regional and global issue as larger proportions of the earth's human population settle in coastal areas (Valiela et al. 1992; de Jonge et al. 2002).

The Chesapeake Bay region of the U.S. has been the focus of much research on the extent and causes of eutrophication (Kemp et al. 2005). Elevated inputs of both nitrogen and phosphorus to the Chesapeake watershed have resulted in excessive phytoplankton production within the Bay (Malone et al. 1986, 1988; Boynton et al. 1982; Correll 1987; Jordan et al. 1991a, b; Gallegos et al. 1992; Harding 1994; Harding and Perry 1997). Consequently, submerged aquatic vegetation has declined (Kemp et al. 1983; Orth and Moore 1983) and hypoxic conditions have increased in both magnitude and extent (Taft et al. 1980; Officer et al. 1984). The detrimental ecological effects of increased nutrient loading to the Chesapeake Bay have led to the multi-state Chesapeake Bay Agreement, which seeks to reduce nutrient discharges to Chesapeake Bay watershed streams and rivers (Boesch et al. 2001).

Nitrogen usually limits primary production in estuarine waters, so that excess N inputs lead to eutrophication (Howarth 1988). In contrast, phosphorus inputs usually limit freshwater primary production (Hecky and Kilham 1988) and may also set the long-term limit on oceanic production (Tyrrell 1999). In estuaries, where freshwater and seawater mix, spatial and temporal changes in the relative availabilities of nitrogen and phosphorus cause shifts in nutrient limitation (Jordan et al. 1991a, b; Doering et al. 1995; Fisher et al. 1999), which complicate the prioritization of nutrient management (Conley 2000).

Nitrogen budgets based on agricultural activities, human populations, and trade of N in food and feed have been successfully applied to understand differences in N loading to aquatic systems (Jordan and Weller 1996; Howarth et al. 1996; Boyer et al. 2002; Van Breemen et al. 2002). These nitrogen budgets quantified the net anthropogenic nitrogen inputs (NANI) into each county by accounting for all inputs and outputs of nitrogen. NANI is nitrogen that is available to accumulate in the landscape, pollute the water, or be released to the atmosphere. New inputs included atmospheric deposition, fertilizer use, biological fixation, and net imports of food and feed. Outputs included gaseous N released to the atmosphere and net exports of food and feed. Similarly calculated county-scale anthropogenic phosphorus budgets would allow the comparison of watershed phosphorus inputs to watershed phosphorus discharges. A comprehensive phosphorus budget, however, has yet to be developed for the entire Chesapeake Bay region.

In this paper, we estimate net anthropogenic phosphorus inputs (NAPI) in the Chesapeake Bay region using the budgeting approach previously developed to estimate NANI (Jordan and Weller 1996; Howarth et al. 1996; Boyer et al. 2002; Van Breemen et al. 2002). NAPI is the net balance of inputs and outputs in a given land area. A positive NAPI implies that phosphorus inputs are greater than outputs, while the reverse is true of a negative NAPI. A positive NAPI represents phosphorus that can accumulate in the landscape and contribute to elevated aquatic phosphorus levels through leaching, erosion, or point source discharges. Thus, NAPI is an index of phosphorus pollution potential.

We use NAPI to quantify the net P inputs to the Chesapeake Bay region and their change through time. We estimate the contributions of specific land uses to NAPI, and we quantify variability in NAPI and P sources among states and geographic provinces. We also assess the potential of simple land cover proportions or population density to act as proxies for the more complex NAPI calculations. We analyze the utility of NAPI as a predictor of P discharges by comparing watershed NAPI estimates to measured P discharges from nine major river basins monitored by USGS. We also quantify the relationships between NAPI, land cover percentages, and phosphorus loads; and we quantify the percentage of inputs that are discharged from nine monitored rivers and from the entire Chesapeake Bay watershed.

Methods

Study area

We constructed phosphorus (P) budgets for all the counties in the states of Delaware, Maryland, Pennsylvania, Virginia, West Virginia, and the District of

Columbia as well as the 20 counties in New York that are wholly or partially within the Chesapeake Bay watershed. Crop and livestock census data are available at the county scale every 5 years from the U.S. agricultural census (United States Department of Agriculture 2002), thus, we developed county P budgets for 1987, 1992, 1997, and 2002. Much of this data has already been organized and was available from the Mid-Atlantic Regional Water Program (2006) which has used it to work up some preliminary county level phosphorus budgets. This approach adds to those previous budgets by incorporating, among other things, trade of P across county boundaries. Land cover was estimated from the National Land Cover Dataset, which was derived from satellite images acquired around 1990 (Vogelmann et al. 1998) and the RESAC land cover classification system (Varlyguin et al. 2001).

Phosphorus budgets

NAPI was estimated by accounting for the anthropogenic phosphorus inputs and outputs in each county. Phosphorus inputs include fertilizer application, net imports of food and feed, and non-food household use of phosphorus (Fig. 1). Phosphorus outputs include net exports of food and feed. Phosphorus transfers among crops, livestock, and humans within each county are incorporated into the budget to assess the contribution of each component to NAPI. Animal waste (manure) is not considered a new input since it is ultimately derived from fertilizer use or imports of feed. Major steps in calculating NAPI include quantifying phosphorus in fertilizer inputs, crop and pasture harvests, animal and human consumption, animal production, trade of feed and food, and non-food inputs.

To more clearly attribute NAPI to particular anthropogenic activities and land uses, NAPI was divided into five components associated with row crop land, pasture land, developed land, enclosed animal facilities, and transfer sinks; and then we assessed the relative magnitudes of these five components at county and regional scales. For any county, these five components sum to the net gain or loss for the entire county, i.e. the total NAPI value. Row crop P inflows include a portion of the county fertilizer application and a portion of the manure from enclosed animal facilities. Row crop outflows include crop P harvested for human and livestock consumption (Fig. 1). Pasture P inflows include manure from grazing animals and enough of the county fertilizer application to balance P removal in animal products. The only pasture outflow is the portion of livestock P production supported by grazing in pastures. Developed land P inflows include imports of food, the portion of crop harvests that is available for human consumption, livestock products from both pastures and enclosed facilities, and P in non-food products, while outputs include any exports of food. Enclosed animal facility P inflows include imports in feed and the portion of crop harvest available for livestock consumption, while P outflows include exports of feed, manure that is applied to row crops, and livestock products. The storage and transfer sink component of NAPI includes P that becomes agriculturally unusable during the transfer and storage of agricultural products. To correspond with Jordan and Weller (1996), we assumed that 10% of food and feed crops are lost in storage and processing based on the percentage of cereal crop lost to pests in storage (Pimentel et al. 1975) and that 20% of hay and silage are lost due to less efficient handling than grains. Thus, transfer sink P inflows include 10 or 20% (depending on crop type) of the crops harvested for consumption by both livestock and humans plus 10% of livestock P produced for consumption by humans.

Phosphorus fertilizer application

Large quantities of fertilizer are applied in agricultural areas and represent a major source of new phosphorus. Fertilizer applications were estimated from county sales data as compiled by the Tennessee Valley Authority (Terry and Kirby 1987–2003). As in previous publications, we assumed that fertilizer is used in the county where it is purchased (Jordan and Weller 1996). Total P fertilizer applications in tons P_2O_5 were converted to kg P by multiplying by 436.4 kg P per ton P_2O_5.

Net P import or export in human food and animal feed

Human and livestock food and feed needs can be met by local agricultural production or by food and feed imports from other areas. With a few exceptions, we

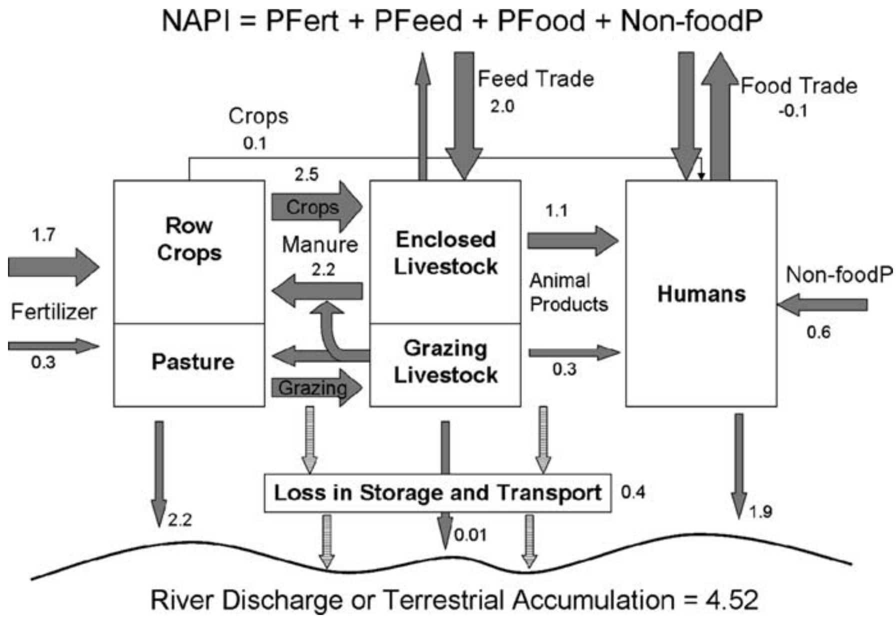

Fig. 1 Flow of phosphorus through agricultural and human reservoirs in the Mid-Atlantic region (including counties in NY state that overlap the Chesapeake Bay watershed). Numbers are average fluxes (kg P ha^{-1} $year^{-1}$)

estimated net phosphorus imports in food and feed following the methods of Jordan and Weller (1996), who estimated net import of nitrogen in food and feed as nitrogen production in crops and animal products minus nitrogen consumption by humans and animals. We obtained crop and animal production data from the U.S. Department of Agriculture, National Agricultural Statistics Service (United States Department of Agriculture 2002), and human population counts from the U.S. Census Bureau for 1990 and 2000 (U.S. Department of Commerce 1990, 2000). Human consumption demands and livestock consumption were compared to available P and any difference was allotted to trade in food and feed respectively.

Food and feed consumption and non-food P

Our current method of estimating consumption by animals and humans is slightly different from our earlier method of using recommended food consumption rates (Jordan and Weller 1996). Humans and livestock often consume more nutrients than recommended, so we instead estimated human and livestock consumption of P by dividing the average amount of P excreted for each animal by the fraction of dietary P that is excreted by each animal (Table 1). This calculation takes into account overfeeding by animals and humans. The P consumption per individual was then multiplied by the number of each animal type in each county as reported in the U.S. Department of Agriculture, National Agricultural Statistics (United States Department of Agriculture 2002). Household sewage discharge was estimated to be only 45% derived from human excretion of P, with the other 55% coming from non-food phosphorus uses such as laundry, bathing, and food wastes from garbage disposals (USEPA 1980; MPCA 2004).

Crop production

Row crop and pasture plants take up P from the soil and make it available for livestock or human consumption. To estimate the total amount of plant P produced in each county, each crop's reported harvested units (United States Department of Agriculture 2002) for the years 1987, 1992, 1997, and 2002 were converted into kg P harvested using the factors in Table 2. Pasture-produced P available for grazing livestock consumption was estimated using acres of pasture land (United States Department of Agriculture 2002) and an average pasture P production of 2.5 kg P ton^{-1} forage and 2 tons forage per acre of pasture (High, Ohio ag. extension special circular 156). The fraction of each crop type's biomass that is produced for consumption by livestock or humans was calculated using the percentages listed in Wedin et al. (1975).

Table 1 Livestock and human phosphorus excretion rates and percentages of dietary P excreted

Animal type	Excretion[a] (kg P individual^{-1} year^{-1})	References	Percent excreted	References
Hogs and pigs	3.03	1, 2	73	Shen et al. (2002)
Milk cows	28.15	1, 2	69	Spears et al. (2003)
Beef cows	16.06	1, 2, 3	62.5	Thomas and Gilliam (1977)
Other cattle	9.11	2, 3	62.5	Thomas and Gilliam (1977)
Layers	0.18	1, 2, 3	87	Iserman (1990)
Pullets	0.16	2	87	Iserman (1990)
Broilers	0.12	1, 2, 3	60	NRC (1994)
Turkeys	0.35	1, 2, 3	71	Branson et al. (1973)
Sheep and lambs	1.07	3, 4	66	Louvandini and Vitti (1996)
Horses and ponies	4.80	1, 2	60	NRC (1989)
Humans	1.00	5	35	Lemann (1996); Nordin (1989)

[a] Adapted from Nutrient Budgets for the Mid-Atlantic States http://mawaterquality.agecon.vt.edu/default.html

1. ASAE (2004)

2. Lander et al. (1998)

3. Service (1993)

4. Van Dyne and Gilbertson (1978)

5. Siegrist and Bolyle (1976), Maybeck et al. (1989), Henze (1997), Libra (2004), Mckee and Eyre (2000), and Strauss (2000)

Grazing livestock

Cows, sheep, and horses can meet some of their dietary needs through grazing on pastures. The fraction of the diet of each animal type that comes from grazing is a function of the availability of pasture crop to graze upon. Availability of large pasture areas in a county reduces the need to feed grazing livestock from row crop production or feed imports. The fraction of a grazing animal's diet that comes from pastures is used to estimate the portion of animal production that is derived from pastures and enclosed facilities. Quantifying the amount of grazing livestock diet that comes from pastures also provides estimates of the amounts of P in manure that are deposited in pastures and collected from animal enclosures to be spread on row crop fields. This helps determine the amount of fertilizer needed to balance P losses from pasture, which in turn modifies the amount of fertilizer applied to row crops.

Pasture production consumed by grazers

Pasture P that is consumed by grazing livestock is estimated using methods first presented in Jordan and Weller (1996) to estimate nitrogen consumption from grazing. To estimate how much P was grazed by cattle, horses, and sheep from pastures; we first calculate the total amount of silage and hay P produced in each county and subtract that amount from the county's livestock consumption demand (this assumes that hay and silage are all consumed by livestock in that same county, Jordan and Weller 1996). Next we estimate the amount of concentrated feed consumed using typical proportions of concentrated feed in grazer diets: 25% for beef cattle, 40% for dairy cows, and 11% for horses and sheep (Hodgson 1978; Wedin et al. 1975), and we subtract this from the remaining livestock consumption demands in each county. Any remaining consumption demands of grazers were assumed to be met by grazing on pasture within the county, but livestock grazing P consumption was limited to the maximum pasture P production for each county. If there were still unmet dietary P demands after assuming grazing at the maximum pasture P production, then these additional demands were assumed to be met by feeding concentrate feeds at higher than typical proportions. This calculation estimates the increased requirements of concentrated feed consumption in areas with scarce pastures. This can accommodate increased concentrated feed demands in intense

Table 2 Agricultural crop production of phosphorus

Crop type	Yield unit	Mid-Atlantic[a] (kg P yield^{-1})	References
Corn grain	Bushel	0.08	1, 2, 3, 4, 5
Corn silage	Ton	0.83	2, 3, 5
Sorghum grain	Bushel	0.08	2, 3, 4
Sorghum silage	Ton	1.11	2, 3
Barley	Bushel	0.08	1, 2, 3, 4, 5
Buckwheat	Bushel	0.07	2
Oats	Bushel	0.07	2, 3, 4, 5
Rye	Bushel	0.09	2, 4
Wheat	Bushel	0.10	1, 2, 3, 4, 5
Alfalfa Hay	Ton	2.95	2, 3, 4
Soybeans	Bushel	0.18	1, 2, 3, 4, 5
Cotton	Bales	1.29	2, 4
Peanuts	Pounds	0.001	6
Potatoes	Hundred weight	0.03	1, 2, 4
Sweet potatoes	Hundred weight	0.01	2
Tobacco	Pounds	0.002	2, 4
Apples	Acres	1.02	7
Sweet cherries	Acres	0.63	7
Grapes	Acres	0.67	7
Peaches	Acres	0.70	7
Pears	Acres	1.32	7
Plums	Acres	0.46	8
Vegetables	Acres	3.28	4, 5
Pasture	Acres	5.00	9

[a] Adapted from nutrient budgets for the Mid-Atlantic States http://mawaterquality.agecon.vt.edu/default.html

1. Sims and Campagnini (2002)
2. Steinhilber et al. (2002)
3. Beegle (2001)
4. Mullins (2005)
5. Rayburn and Basden (2004)
6. Lander et al. (1998)
7. USDA Statistical Highlights (2001)—crops
8. USDA National Nutrient Database for Standard References (2005)
9. High, Ohio agricultural extension special circular 156 and various other agricultural extensions

agricultural areas that rely on feedlot production. Removal of P from pastures in livestock biomass exported for human consumption was balanced by application of a portion of the fertilizer bought in each county. Thus, less fertilizer is allocated to row crops in counties with large pastoral livestock production.

Livestock products for human consumption

Humans consume P in both plant and animal products. Livestock production of P that is available for human consumption was estimated using livestock numbers from the agricultural census (United States Department of Agriculture 2002) and the difference between consumption and excretion of P for each animal (Table 3). This may be an overestimate since some of the assimilated P will end up in bones and other animal parts that are non-edible to humans.

Geographic and temporal analysis

Spatial patterns in NAPI and the five land-cover components were examined by grouping counties into four dominant land use categories using data on land cover, the intensity of livestock production, and human population density. The NLCD land cover data set (Vogelmann et al. 1998) was used to calculate, with ArcInfo 9.2 (ESRI Inc.) geographic information system, the proportions of row crops and forest in each county, the average number of livestock units was calculated from agricultural census livestock inventories between 1987 and 2002, and average human population densities were calculated from the 1990 and 2000 U.S. Census. A livestock unit is 1,000 kg of animal weight which, for example, is equivalent to approximately 250 egg laying chickens, 5.88 hogs, or 1 beef cow. The four dominant land use categories were defined as: counties with >70% forest land, >10% row crop land, >0.3 livestock units per hectare, and >3 people per hectare. We also examined differences

Table 3 Animal food phosphorus production per year for common livestock

Livestock type	Product	kg P year^{-1} individual^{-1}
Milk cows	Milk	8.73
Layer chickens	Eggs	0.02
Beef cows	Meat	6.02
Broiler chickens	Meat	0.05
Turkeys	Meat	0.10
Hogs and pigs	Meat	0.82
Sheep and lambs	Meat	0.36

Animal production is calculated by subtracting the amount of P excreted (see Table 1) from the amount of P consumed in NAPI and its components among states and physiographic provinces. Temporal trends in NAPI and its components between 1987 and 2002 were assessed by comparing regional and county level estimates of NAPI among years.

Proxy variables for NAPI

To evaluate the feasibility of using land cover percentages, livestock density, or human population density as proxies for the more intensive NAPI calculations, we used step-wise linear regression to fit empirical models predicting NAPI from the potential proxy variables. Variables were log transformed to address issues of non-normality in the data.

Discharge of NAPI

Phosphorus inputs to the landscape can be captured in useful agricultural products (such as plant biomass), can accumulate in soils, or can be discharged into streams. The percentage of inputs that is discharged has important implications for watershed management of nutrient loads. The total mass of net anthropogenic P in the Chesapeake Bay watershed was calculated by summing across counties the area of each county located within the drainage basin times the county NAPI. This estimate of the mass of P available to drain to the Chesapeake Bay was then compared to the Chesapeake Bay Program's Watershed Model estimate of 8.66 million kg of P discharged annually to the Chesapeake Bay (CBP 2006). The ratio of terrestrial P inputs and average P loads discharged to Chesapeake Bay was used to estimate the percentage of NAPI that is discharged into the Chesapeake Bay or that builds up in the landscape.

Spatial variability in phosphorus inputs to the landscape as well as topographic, geological, and hydrological differences among sub-watersheds of the Chesapeake Bay watershed may influence the relationship between NAPI values and measured P loads. Loads of nutrients from USGS River Input Monitoring (RIM) watersheds have already been computed by USGS (Langland et al. 2006) using the 7-parameter, log-linear regression model (ESTIMATOR) developed by Cohn et al. (1989). NAPI was calculated and apportioned, using ArcInfo 9.2 (ESRI Inc.) geographic information system, into nine RIM station watersheds (Langland et al. 2006; RIM 2004) to assess large scale differences in the proportion of NAPI that is discharged in streams. County NAPI was apportioned, using ArcInfo 9.2 (ESRI Inc.) geographic information system, to the watersheds using the fraction of each county that intersects a given watershed. Enclosed animal facility and storage and transport sink NAPI components were also apportioned by percent of county area within each watershed. Developed and row crop NAPI components were apportioned from counties to watersheds using the ratio of RESAC developed or row crop land cover class (Varlyguin et al. 2001) in a county to developed or row crop land cover class in each watershed county intersect area. The RESAC land cover classification of land cover is specifically designed to differentiate pasture, grasslands, and row crops better than the NLCD classification method (Vogelmann et al. 1998). The RESAC land cover classification for Chesapeake Bay, however, does not cover all the counties in our budget and so could only be used for the apportioning to watersheds part of our research. The NLCD, as mentioned above, was used to summarize the amount of each land cover type present for comparisons between counties and states and for working up proxy variables for the NAPI calculation. Phosphorus loads at USGS gauges draining the RIM watersheds (Langland et al. 2006) were compared to NAPI and its components in each watershed. Watershed characteristics such as land cover percentages were also compared to P discharges to assess the added predictive power of using NAPI.

Results

County characteristics

Landscape characteristics and NAPI values vary widely among counties in the Chesapeake Bay region. Overall the Chesapeake Bay watershed was 64% forest, 19% pasture, 6% row crop, and 4% developed; but NLCD land cover varied considerably among counties. There are counties with intense row crop land cover such as Kent County, Maryland (47% row crop land). Some counties have more than 50% pasture land, such as Carroll in Maryland and Lancaster in Pennsylvania. Some counties are still heavily forested, including the counties of Cameron and Forest in

Pennsylvania, Webster, McDowell and Wyoming in West Virginia, and Buchanan in Virginia; which all have over 95% forest cover. Counties containing over 30% developed land cover include Philadelphia and Delaware in Pennsylvania, Arlington and Newport News in Virginia, and the District of Columbia. Human population densities are correspondingly high in Philadelphia and Delaware counties in Pennsylvania (42 and 11 people ha^{-1}, respectively) as well as Washington, DC (25 people ha^{-1}). Livestock densities are highest in Lancaster, Pennsylvania with 1.2 livestock units ha^{-1}.

Net anthropogenic phosphorus inputs and geographic differences

The different management practices associated with forests, row crops, livestock production, and developed land produce a wide range of NAPI values among counties (Fig. 2). The average of 1987, 1992, 1997, and 2002 NAPI values for individual counties in this study ranged from 0.02 to 78.46 kg P ha^{-1} year^{-1}. The overall area-weighted average NAPI for the 266 counties within the study area, which includes all of the Mid-Atlantic States plus New York counties that intercept the Chesapeake Bay watershed, between 1987 and 2002 was 4.52 kg P ha^{-1} year^{-1}. Particular land types and activities strongly affected the magnitudes of NAPI and its components (Table 4). The 125 counties with greater than 70% forested lands had the lowest average NAPI at 1.65 kg P ha^{-1} year^{-1}. The 50 counties with greater than 10% row crop land had an average NAPI of 6.82 kg P ha^{-1} year^{-1}. Intense animal production in 12 counties with greater than 0.3 livestock units per ha^{-1} yielded a NAPI of 15.8 kg P ha^{-1} year^{-1}. High human population densities above 3 people ha^{-1} in 16 counties yielded the largest average NAPI of 19.62 kg P ha^{-1} year^{-1}.

Differences in land use led to differences in NAPI among the states (Table 5). The District of Columbia is highly urbanized and has a NAPI of 44.30 kg P ha^{-1} year^{-1}. Delaware has a NAPI of

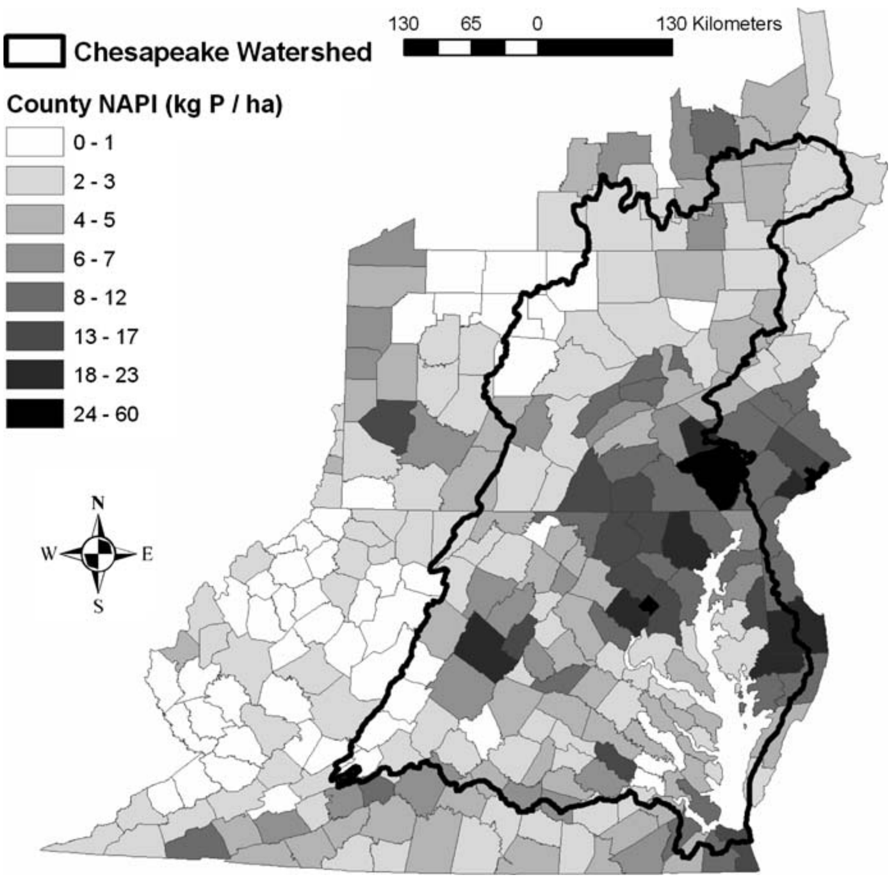

Fig. 2 Average net anthropogenic phosphorus inputs in the counties of the Mid-Atlantic region and part of New York State for 1987, 1992, 1997, and 2002. The black outline is the boundary of the Chesapeake Bay watershed

Table 4 Area standardized average county net anthropogenic phosphorus input components (kg P ha^{-1} year^{-1}) grouped by land cover, livestock unit (LU) density, and human population density

County group	*N*	Row crop NAPI	Developed NAPI	Loss in storage and transport NAPI	Total NAPI
All	266	2.19	1.91	0.41	4.52
>70% Forest	125	0.83	0.60	0.22	1.65
>10% Row crop	50	4.55	1.54	0.72	6.82
>0.3 LU ha^{-1}	12	12.74	1.52	1.52	15.80
>3 People ha^{-1}	16	2.07	17.32	0.23	19.62

Enclosed Animal Facility NAPI < 0.02 is omitted from table

Table 5 Average county human population (ha^{-1}), livestock unit (LU) density (ha^{-1}), and net anthropogenic phosphorus input intensities (kg P ha^{-1} year^{-1}) in each state of the Mid-Atlantic and the part of New York State within the Chesapeake Bay Drainage

State	Pop	LU	Row crop NAPI	Developed NAPI	Loss in storage and transport NAPI	Total NAPI
DC	25.23	0.00	0.000	44.30	0.00	44.30
DE	1.79	0.18	9.21	2.54	1.28	13.03
MD	1.55	0.13	4.97	3.42	0.66	9.06
NY	0.51	0.15	1.45	0.87	0.60	2.93
PA	1.69	0.14	2.62	3.10	0.53	6.26
VA	0.88	0.11	2.58	1.16	0.37	4.12
WV	0.36	0.06	0.42	0.53	0.14	1.09
All			2.19	1.91	0.41	4.52

Enclosed Animal Facility NAPI < 0.02 is omitted from table

13.03 kg P ha^{-1} year^{-1} and Maryland has a NAPI of 9.06 kg P ha^{-1} year^{-1}, and row crops are the largest component of NAPI in both states. Pennsylvania, Virginia, and the part of New York that intersects the Chesapeake Bay watershed have lower NAPI values with moderate row crop and urban P inputs. West Virginia has the lowest NAPI at just 1.09 kg P ha^{-1} year^{-1}.

Among physiographic provinces, the Coastal Plain and the Piedmont have NAPI values more than double those of either the Appalachian mountain or Plateau provinces (Table 6). The Appalachian plateau province had the lowest NAPI at 2.26 kg P ha^{-1} year^{-1}. The Piedmont province had the highest NAPI at 9.11 kg P ha^{-1} year^{-1}.

Temporal changes

Overall there has been a non-significant but slight upward trend in NAPI between 1987 and 2002 (linear

Table 6 Average county net anthropogenic phosphorus input intensities completely contained by each physiographic province within the Chesapeake Bay watershed (kg P ha^{-1} year^{-1})

Province	Row crop NAPI	Developed NAPI	Loss in storage and transport NAPI	Total NAPI
Appalachian Plateau	1.20	0.56	0.50	2.26
Appalachian Mountain	2.04	0.85	0.43	3.33
Coastal Plain	3.17	2.29	0.47	5.93
Piedmont	5.94	2.37	0.78	9.11
All Chesapeake Bay				5.41

Total number of counties equals 156. Enclosed Animal Facility NAPI < 0.02 is omitted from table

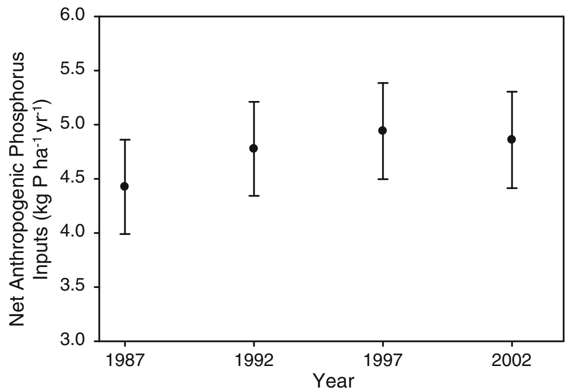

Fig. 3 Temporal changes in average county major net anthropogenic phosphorus input components. Error bars are ±one standard error ($n = 266$)

regression of NAPI versus year, $p = 0.17$). County area-weighted NAPI increased from 4.43 to 4.94 kg P ha^{-1} year^{-1} between 1987 and 1997 but decreased slightly to 4.86 kg P ha^{-1} year^{-1} by 2002 (Fig. 3).

Phosphorus fluxes

Our budget model (Fig. 1) accounts for gross P fluxes within a county and net fluxes across county boundaries. The net fluxes represent gains or losses from a county. We calculated average values of several of the gross and net fluxes for the entire Mid-Atlantic region plus New York counties that intercept the Chesapeake Bay watershed (Figs. 1 and 4). The largest gross phosphorus fluxes were associated with row crop and livestock agriculture. Crops grown for livestock consumption and manure applied to row crops are the two largest gross fluxes of phosphorus. These are followed by livestock production in enclosed animal facilities, livestock production from pastures, and finally row crop production for human consumption. Trade in feed for livestock consumption and fertilizer applications to row crops make up the two largest net fluxes of phosphorus. The next two largest net fluxes of phosphorus are associated with non-food phosphorus use, and fertilizer applied to pasture land. Trade of food for human consumption is the only net negative phosphorus flux.

Proxy variables

A few easily calculated proxy variables explained much of the variability in NAPI among counties. Human

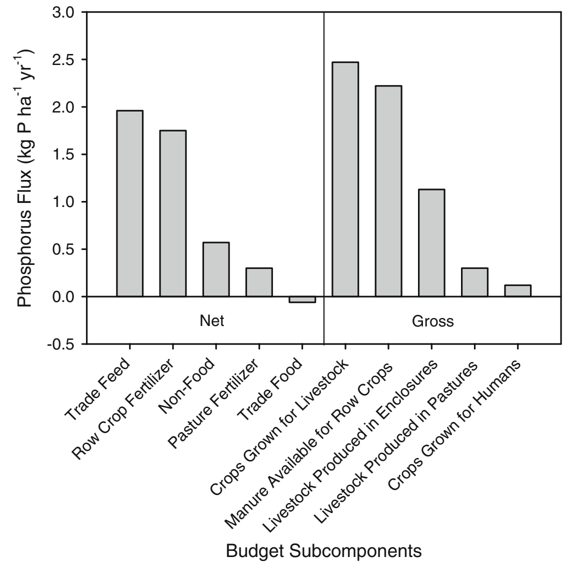

Fig. 4 Annual average budget phosphorus fluxes (1987, 1992, 1997, and 2002) for counties in the Mid-Atlantic region (including counties in NY state that overlap the Chesapeake Bay watershed). External components are used for the calculation of net anthropogenic phosphorus inputs. Internal components represent fluxes of phosphorus among row crops/ pasture, livestock production, and human reservoirs

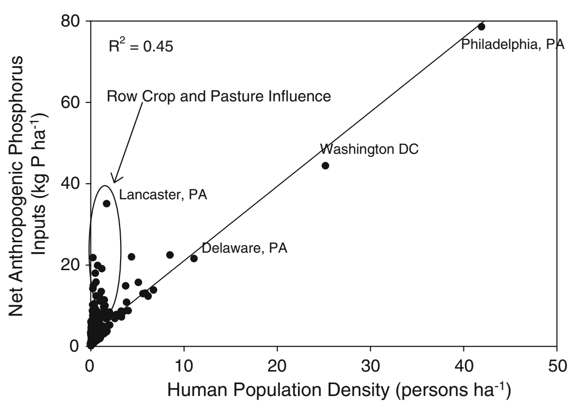

Fig. 5 Net anthropogenic phosphorus inputs relationship with human population density. R^2 based on log transformed variables. A few agriculturally based counties (circled) have very high NAPI even at lower proportions of developed land

population density (log transformed) was the best single significant predictor of NAPI (log transformed) (linear regression, $R^2 = 0.45$, $p < 0.0001$) (Fig. 5). NAPI increased gradually from 0 kg P ha^{-1} year^{-1} in counties with low population densities to 40–80 kg P ha^{-1} year^{-1} in counties such as Philadelphia, PA and Washington, DC that have large urban centers. After accounting for human population density (step-wise linear regression), livestock unit density (log transformed) is the second best predictor of NAPI (partial $R^2 = 0.32$, $p < 0.0001$). NAPI increased from values near zero, in counties with almost no livestock production, to around 35 kg P ha^{-1} year^{-1} in counties with very intense livestock production such as Lancaster, PA. NAPI also increased from close to 0 kg P ha^{-1} year^{-1} in counties with little to no row crop land to around 10–20 kg P ha^{-1} year^{-1} in counties with row crop land greater than 10%. Human population density and livestock unit density (log transformed) were better predictors of NAPI (Cumulative $R^2 = 0.77$) than percent developed land and percent pasture land (log transformed) (Cumulative $R^2 = 0.53$). A combination of independent variables including human population density, livestock unit density, and percent row crop land explained 83% of the variability in NAPI when analyzed using step-wise linear regression after log transformations ($p < 0.0001$). An average hectare of row crops has a NAPI of 2.19 kg P ha^{-1} year^{-1} while developed land has 1.91 kg P ha^{-1} year^{-1}.

Percentage of NAPI discharged from Chesapeake Bay watersheds

After apportioning NAPI from counties into the areas of the Chesapeake Bay watershed we compared NAPI to phosphorus discharges from the landscape. Our calculation of average NAPI in the 175 counties located at least partly within the Chesapeake Bay drainage basin was 5.41 kg P ha^{-1} year^{-1} which is higher than the overall study area average of 4.52 kg P ha^{-1} year^{-1} due to higher concentrations of human and agricultural land use within the borders of the Chesapeake Bay drainage. The Chesapeake Bay Program's Watershed Model (CBP, Chesapeake Bay Program 2006) estimated a discharge of 8.7 million kg P year^{-1} from the 167,000 km^2 watershed to the Chesapeake Bay in 2000 (0.52 kg P ha^{-1} year^{-1}) and the USGS river input monitoring project estimated a long-term average P discharge of 9.2 million kg P year^{-1} (0.55 kg P ha^{-1} year^{-1}; Langland et al. 2006; RIM 2004). The P yield of 0.52 kg P ha^{-1} year^{-1} discharged to the Chesapeake Bay is about 10% of NAPI (5.41 kg P ha^{-1} year^{-1}) leaving 90% of the NAPI to potentially accumulate in the landscape.

The nine watersheds in the USGS river input monitoring program collectively represent 93% of the water draining from the nontidal part of the Chesapeake Bay and approximately 73% of the combined area of the Chesapeake Bay and its watershed. Land use percentages within the watersheds (Table 7) range up to 81% forest in the James River watershed, to 29% row crop land in the Choptank watershed, 25% developed land in the Patuxent River watershed, and 30% pasture in the Rappahannock watershed. NAPI was lowest in the Mattaponi River watershed at 1.89 kg P ha^{-1} year^{-1}, and highest in the Choptank watershed at 10.20 kg P ha^{-1} year^{-1} (Table 7). Watershed total P discharges ranged from 0.03 to 0.40 kg P ha^{-1} year^{-1}. Total NAPI explained 34% of the variability in watershed discharges with a trend of increased discharge with increased NAPI values (Fig. 6). The relationship was not statistically significant however ($p = 0.10$). And we can only estimate from the slope of the trend lines that, overall, about 2% of NAPI is discharged from these watersheds. Certain NAPI components had a more defined relationship with watershed P discharges than total NAPI. The developed component of NAPI, after log transformation, explained 81% of the variability in P discharges (linear regression, $p = 0.0009$), but the same amount of variability in total P discharges was explained by just using percent developed land ($R^2 = 0.82$, $p = 0.0008$) which does not account for the intensity of P inputs into different developed land parcels. A combination of percent developed land and human population density, after log transformations, explained 95% (step-wise linear regression, $p < 0.0001$) of the variability in total phosphorus discharges from the RIM watersheds.

Discussion

Our calculation of NAPI and various NAPI components linked to different land uses allowed us to estimate the amount of excess phosphorus in the landscape that has potential to pollute. Across counties, NAPI was found to be generally positive, which we expected due to inefficiencies in agricultural use of fertilizers, manure applications etc. and from past research showing that, in some areas, intense anthropogenic land use can lead to soil conditions that are becoming saturated with phosphorus (Mallarino et al. 1991; McCollum 1991). Research is enhancing knowledge of the detrimental effects of this excess

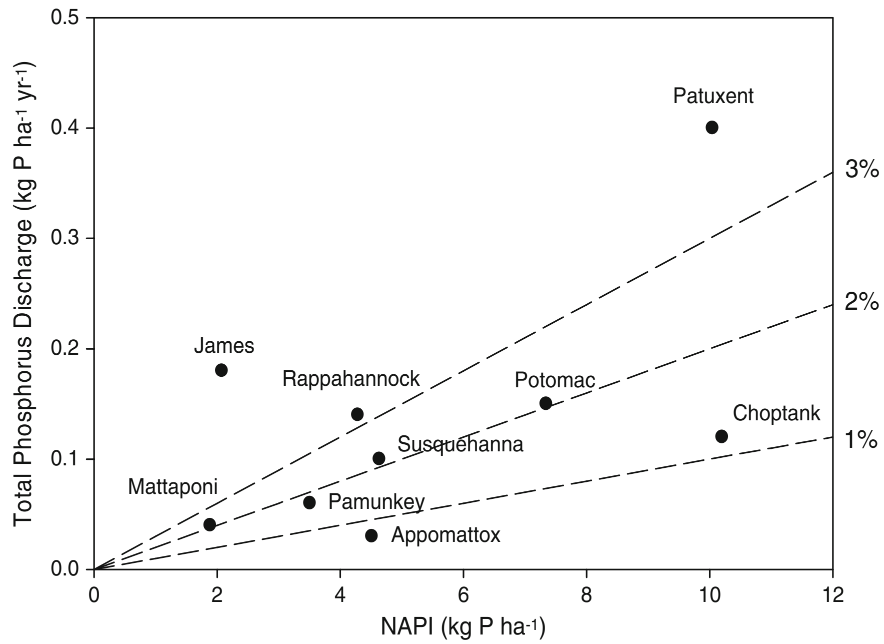

Fig. 6 Total phosphorus discharge related to net anthropogenic phosphorus inputs in USGS river input monitoring watersheds. About 2% of inputs are discharged from these watersheds

phosphorus on stream systems (Sharpley et al. 1994; 1996, Sharpley 1995; Shepherd 2000). Phosphorus loads, however, continue to be logistically difficult to measure because they tend to be delivered in particulate form during sediment pulses (Staver and Brinsfield 1995; Coale 1999). The results of this study establish spatially explicit baseline phosphorus budgets for input into models linking NAPI to watershed discharges in currently unmonitored watersheds in the Chesapeake Bay region.

Our estimate of average county scale NAPI between 1987 and 2002 is 4.52 kg P ha^{-1} and at the state scale ranges between 1.09 and 13.03 kg P ha^{-1} year^{-1} if the highly urbanized Washington, DC (44.30 kg P ha^{-1} year^{-1}) is not included. Other budget studies estimate terrestrial P inputs of 0.092 kg P ha^{-1} year^{-1} in Victoria, Australia (Campbell 1978), 9.7 kg P ha^{-1} year^{-1} in the upper Potomac, USA (Jaworski et al. 1992), and 11.5 kg P ha^{-1} year^{-1} in Lac Leman, France (Pilleboue and Dorioz 1986). Our regional budget, thus, falls within the range of other estimates of phosphorus inputs in different landscapes.

Decoupled production and consumption

One effect of the large human population and associated agricultural practices in the Chesapeake Bay region has been the gradual build up of phosphorus in the landscape (Boesch et al. 2001). The decoupling of P production and consumption at the county scale in the Chesapeake Bay landscape results in positive net anthropogenic phosphorus inputs (NAPI) in most counties (Fig. 2). County level NAPI depends largely on the relative intensities of different uses of phosphorus within each county. The less efficiently an agricultural source of phosphorus is turned into a useable form, the more influence that source will have on the magnitude of NAPI. High demands for food P and little agricultural production capability in urbanized counties have a trickle down effect on their hinterlands, contributing to intense row crop and animal production in counties that, themselves, do not have much urban development. Row crop and animal production areas, in turn, often require their own imports of phosphorus in fertilizer and feed from outside county boundaries to meet demands that cannot be met from local P reservoirs. Thus, the actual anthropogenic impact from P consumption by humans is spread throughout our budget and in space. The relationship (Fig. 5) between human population densities and NAPI (linear regression, $R^2 = 0.45$, $p < 0.0001$) is not surprising considering the large human demands of P from crops and livestock, and the current inefficiency of returning P waste products to the agricultural

Table 7 River monitoring watershed average net anthropogenic phosphorus input components (kg P ha^{-1} year^{-1})

Station number	Watershed	Percent developed	Percent row crop	Pop	LU	Row crop NAPI	Developed NAPI	Loss in storage and transport NAPI	Total NAPI	Phosphorus discharge	Percent discharge
1491000	Choptank	5.20	29.13	0.63	0.17	6.30	0.73	1.21	10.20	0.12	1.18
1578310	Susquehanna	4.22	7.82	0.55	0.18	2.98	1.00	0.63	4.64	0.10	2.16
1594440	Patuxent	25.38	7.32	4.22	0.06	1.74	6.87	0.34	10.05	0.40	3.98
1646580	Potomac	5.43	7.03	0.67	0.26	5.31	1.25	0.73	7.35	0.15	2.04
1668000	Rappahannock	1.92	5.68	0.27	0.20	3.25	0.43	0.63	4.29	0.14	3.26
1673000	Pamunkey	1.54	5.10	0.35	0.10	2.25	0.39	0.36	3.51	0.06	1.71
1674500	Mattaponi	2.21	9.29	0.34	0.04	1.08	0.46	0.21	1.89	0.04	2.12
2035000	James	1.67	1.50	0.17	0.10	1.07	0.44	0.27	2.08	0.18	8.65
2041650	Appomattox	1.11	4.33	0.30	0.13	3.63	0.29	0.38	4.52	0.03	0.66

Enclosed Animal Facility NAPI <0.02 is omitted from table. Average county human population (Pop) (ha^{-1}), livestock unit (LU) density (ha^{-1})

sector for re-use. Human population densities, livestock production intensity, and percentage of land devoted to row crops, thus, serve as good predictors of NAPI at the county scale and when combined explain 83% (step-wise linear regression, $p < 0.0001$) of the variability in NAPI at the county scale.

Trade of feed and food

Trade of feed is a major pathway for phosphorus inputs to the Chesapeake Bay region. Dense urban populations and areas of intensive animal production drive that trade. On average, trade of feed is the largest net P flux followed by fertilizers for row crops (Fig. 4). Out of all net P inputs to counties, trade of feed accounts for about 43% of the total. Counties known from previous budgeting attempts (Jordan and Weller 1996; Mid-Atlantic Regional Water Program 2006) to have large agricultural nutrient demands (such as Lancaster, PA) have correspondingly high NAPI values in our study (Fig. 2). The Mid-Atlantic Phosphorus Budget, which this work uses as a foundation, fails to capture the true phosphorus balance in some counties because it does not include trade of feed and food into and out of counties (Mid-Atlantic Regional Water Program 2006). The Mid-Atlantic agricultural based budget is, in essence, assuming that all crop and livestock production remains in the county where it was produced. Thus, areas of intense crop and livestock production tend to have much more excess phosphorus and areas of development tend to have much less excess phosphorus than what we estimate after accounting for trade.

The transfer of phosphorus in the form of fertilizers and manure through agricultural systems to humans can be an inefficient process (Fig. 4). A comparison of overall average fertilizer inputs to average crop production yields phosphorus transfer efficiency rates close to 76%. A comparison of average animal consumption with average animal production, however, yields animal phosphorus production efficiency around 33%. The production of phosphorus in animal products is, thus, by far the less efficient step in the agricultural production of food products for human consumption.

Very little phosphorus in row crop production (<3%) is channeled into plant products consumed directly by humans, which is the more efficient pathway than channeling P through livestock. Of the net sub-components of NAPI, fertilizer applications to row crop and pasture land account for 45% of the total (Fig. 4). Imports of feed to the region account for another 43% of NAPI; leaving non-food phosphorus uses to account for 12% of NAPI. Manure available for application to row crops (2.2 kg P ha^{-1} $year^{-1}$) is more than enough to balance fertilizer P applied to row crops (1.7 kg P ha^{-1} $year^{-1}$) if manure can be efficiently transported to where it is needed. Our budgets balance any outflows from pasture land by applying some of the available fertilizer to pastures. This fertilization of pasture land, to balance phosphorus removal by livestock production, accounts for about 15% of all fertilizer use in our budget. Fertilizer applied to pasture land (0.3 kg P ha^{-1} $year^{-1}$) could be replaced by the left over manure (0.5 kg P ha^{-1} $year^{-1}$) that is not needed to balance row crop fertilizer requirements in this region. All P requirements by row crop and pasture lands could, thus, be met without any use of chemical/inorganic fertilizers through the proper application of manure P produced within the region. This adjustment in waste management would close the regional agricultural nutrient loop for P and could potentially decrease NAPI to the region by up to 45%.

The negative average value for overall trade of food P (Fig. 1) is interesting, since it implies that the agriculturally based counties within the region are sufficient to provide P to the highly urbanized ones. Even though the region as a whole exports food P, areas with high human population densities still tend to have high NAPI's because NAPI depends on the balance between human consumption and the magnitudes of crop and animal production for human consumption within each county. Cities such as Washington, DC and Philadelphia, PA have large net imports of food (as high as 50 kg P ha^{-1} $year^{-1}$) which account for up to 71% of their NAPI's.

Other possible fates of P

Our calculations of the developed NAPI component are probably overestimates, and other components are underestimated, because data were not available to account for some of the fates of P at the county scale. For example, we could not account for that portion of sewage sludge that is removed from waste water treatment plants and transported to landfills or incinerators. State information on the fates of sewage sludge is available, but information about the fates of sewage sludge at the scale of individual counties is not readily available. The percentage of sewage sludge that is disposed of in landfills and incinerators is estimated to be around 11% in Maryland (Maryland Department of the Environment 2006). The other 89% is applied to the land in some form with 50% of the state generated sewage sludge being applied to agricultural land, 18% composted or palletized for commercial soil supplements, and 21% used for land reclamation. In Pennsylvania about 20% of the sewage sludge is sent to landfills (Pennsylvania Department of Environmental Protection 2006). Overall, land applications in any form do not remove phosphorus from a county unless waste is shipped to another county. Therefore, land application should not be accounted as a net outflow in the calculation of NAPI, but rather a flow between subcomponents. Addition of a sewage sludge disposal term would tend to decrease the developed NAPI component and increase the row crop and pasture components as most sludge disposal land would tend to be more agricultural than urban. Little info is also available at the county scale about the fate of animal parts not used for human consumption such as bones and feathers. A 1991 USDA report states that, nationally "approximately 7.9 billion pounds of meat and bone meal, blood meal and feather mean [were] produced in 1983." Of that amount, 34% was used in pet food, 34% in poultry feed, 20% in pig food, and 10% in beef and dairy cattle feed. Thus the transport of P in "non-consumable" animal parts may, for the most part, end up being recycled into animal production, but may also get incorporated into developed areas through consumption of pet food.

NAPI's relationship to nutrient discharges

NAPI values can be indicators of potential phosphorus accumulation or discharge from the landscape. Our calculations of NAPI values in the USGS's river input monitoring watersheds range between 1.89 and 10.20 kg P ha^{-1} $year^{-1}$ (Langland et al. 2006; RIM, River Input Monitoring Program 2004). Other attempts to quantify terrestrial phosphorus inputs in sub-watersheds within the Chesapeake Bay (Boynton et al. 1995) drainage and in other areas of the world (Mckee and Eyre 2000) show similar ranges of phosphorus inputs as our values. Our budgets allow for quantification of NAPI using readily available remotely sensed and census data and serve as a basis for relating excess P inputs to stream nutrient yields. The lack of a statistically significantly relationship ($p = 0.15$) between log transformed NAPI and P discharge in RIM watersheds may be an artifact of the way USGS samples (Fig. 6). USGS's non-flow weighted sampling methods could lead to underestimates of total P loads or estimates that are more related to the number of storm event samples they measure than to actual loads over time.

Total phosphorus loads in river discharges into Chesapeake Bay have been estimated by the Chesapeake Bay Program's Watershed Model (CBP 2006), the USGS's SPARROW model (RIM, River Input Monitoring Program 2004), and in sub-basins of the Chesapeake Bay watershed (Boynton et al. 1995). The CBP and the SPARROW model both estimate similar total P loads to Chesapeake Bay of around 9 million kg P year^{-1} (0.52–0.55 kg P ha^{-1} year^{-1}). Our estimate of total NAPI for the region draining to the Chesapeake Bay is around 90.5 million kg P year^{-1} (5.41 kg P ha^{-1} year^{-1}). Thus, phosphorus discharges to the Chesapeake Bay are only 10% of NAPI, leaving an abundance of P to build up in some form in the landscape. In the RIM watersheds, we estimate that between 0.7 and 8.7% of NAPI is discharged (Table 7). Similarly, the SPARROW model estimates a similar range, between 2 and 10%, of phosphorus inputs are discharged from sub-basins of the Chesapeake Bay watershed (Sprague 2000). Local scale differences between our results and those of SPARROW and CBP may become evident as we begin assessing NAPI's relationship to P discharges in smaller sub-watersheds, such as those in Jordan et al. (1997) and Weller et al. (2003). Smaller watersheds tend to have only a limited sub-set of the types of agricultural and urban activities that get grouped together when looking at larger watersheds. Our break down of NAPI into components that are associated with particular land covers will allow us to more accurately estimate NAPI in these smaller watersheds than past budget efforts could. The SPARROW and CBP models are also constructed so that both fertilizer and manure applications are counted as new net inputs to row crop land (Smith and Alexander 1997; Linker et al. 1999), and both approaches also do not incorporate trade of food and feed to offset excesses or shortages in production at the county scale. We do not count manure as a new net input since it is ultimately derived from food grown in the county using fertilizer inputs or from feed entering the county via trade. Counting manure, fertilizer applications, and feed imports as all being new imports is essentially double counting. Thus, our complete budgeting approach represents an improvement in the distribution of nutrients among various agricultural and urban sectors.

Boynton et al. (1995) estimates a slightly higher load to the entire Chesapeake Bay drainage area than both models mentioned above. Their estimate is around 11.25 million kg P year^{-1}, but this estimate is based on nutrient loads coming from more urbanized coastal plain watersheds. This loading estimate would increase our calculation of the percentage of NAPI discharged to Chesapeake Bay to around 12.5%. This higher value may be more relevant for urbanized coastal plain watersheds since estimates of discharged loads using SPARROW and CBP models are derived from measurements taken at monitoring stations upstream of many of the coastal urbanized areas. Point sources associated with the urbanized coastal plain section of the Chesapeake Bay watershed may be responsible for the higher proportion of inputs being discharged there. In urbanized watersheds, point sources can account for greater than 60% of all phosphorus discharges (Nemery et al. 2005). Point source discharges efficiently transfer phosphorus inputs to streams and P discharges can be greater than 25% of P inputs to streams in urbanized areas (McMahon and Woodside 1997). In the Albemarle-Pamlico drainage basin, North Carolina and Virginia, it was estimated that even though permitted P point source inputs from waste water treatment plants and industry accounted for 16% of the P source inputs to the watershed, they accounted for greater than 40% of the stream loads (Spruill et al. 1998).

The low percent discharge of most terrestrial P inputs is not surprising, especially where diffuse P inputs dominate, since P tends to bind to sediments (Weiskel and Howes 1992). Reservoirs also can trap a high proportion of phosphorus from the landscape. The Conowingo Reservoir on the lower Susquehanna River, for example, is currently trapping about 40% of the phosphorus delivered from the Susquehanna watershed (Sprague 2000). Percent P discharge, however, may well depend on local soil conditions, riparian buffer retention capacity, and the intensity of phosphorus inputs. Correll et al. (1992) estimated that only 48% of phosphorus inputs were retained in the landscape of the Rhode River tidal basin where 93% of the inputs come from farm chemicals. In areas with riparian buffers up to 80% of phosphorus inputs were retained (Peterjohn and Correll 1984). A global model of river-borne dissolved inorganic phosphorus export called NEWS-DIP predicted that of the 34 Tg of P year^{-1} input onto watersheds by human activity globally, approximately 3% reaches river mouths as DIP (Harrison et al. 2005). Anthropogenic sources account for 65% (0.71 Tg year^{-1}) of the DIP exported to the coastal zone, with the remainder (0.38 Tg year^{-1}) attributable to natural weathering processes; DIP yields range over five orders of magnitude, from <0.01 to 1,153 kg P km^{-2} year^{-1} with highest predicted DIP yields clustering in East Asia, Europe, and Indonesia. At both regional and global scales, the high retention of phosphorus inputs in the landscape is causing soil accumulation levels that will have long reaching effects on eutrophication of freshwater and coastal ecosystems (Bennett et al. 2001).

Human sewage is the largest anthropogenic source of DIP to the coastal zone on all continents and to all ocean basins (Harrison et al. 2005). NEWS-DIP also suggests that despite regional variability, at the global scale, non-point sources of DIP such as inorganic P fertilizer and manure are much less important in determining coastal export of DIP than point sources and natural weathering processes. In agreement with the relative importance of point versus non-point source discharges, much of the variability (81–82%) in nutrient discharges from the river monitoring watersheds in this study was explained by either the developed NAPI component or by the percentage of developed land where point sources tend to be more prevalent. Globally, urban human population density, which corresponds to both developed land and the intensity of P inputs to that land, has been shown to be a good predictor ($p < 0.0001$, $R^2 = 0.59$) of P export from 32 large rivers around the world (Caraco 1995). At the large scale of these watersheds, the intensity of P inputs to each land type may not be as important as the amount of that land type. The larger a watershed gets the less influence rare cases of land use have as their inputs are diluted by an abundance of other land use areas. As already mentioned, our NAPI budget and its land cover linked components may be able to explain more of the variability than just land-cover percentages in smaller watersheds where spatial heterogeneity of the landscape becomes more prevalent.

Temporal trends

The slight increase (12%) in NAPI between 1987 and 2002 (Fig. 3) follows the general population increase in the Chesapeake Bay region. The Chesapeake Bay Program (CBP, Chesapeake Bay Program 2006) reported a 28% human population increase between 1970 and 1997, with an estimated 300 new people moving to the region every day. The inter-annual variability in NAPI is greatly influenced by fertilizer inputs, and the balance of feed and food trade, which do not follow meaningful patterns between 1987 and 2002.

Conclusion

Eutrophication of Chesapeake Bay waterways continues to be a major focus of management and scientific efforts. This study, to our knowledge, is the first time phosphorus budgets have been calculated so thoroughly on a regional scale. The relationships between landscape characteristics (land cover, livestock, and human population densities) and NAPI found in this study may prove invaluable for quick estimation of NAPI in other areas of the county. This study serves as the foundation for further exploration of anthropogenic effects on phosphorus inputs at the landscape scale.

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